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Use of the index of biotic integrity to assess physical habitat degradation in warmwater streams

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Abstract

Indices of biotic integrity (IBI) were computed for two annual fish collections from 27 locations along the bluffline bordering the Mississippi River alluvial plain in northwestern Mississippi. Study sites exhibited varying degrees of physical habitat degradation due to accelerated channel erosion. Objectives of index application were to quantify existing environmental quality and to test the IBI as a tool for relating fish population characteristics to physical degradation. Physical habitat data were collected concurrently with fish at all sites, and physical habitat descriptors were compared with the IBI scores and component metrics.

Three to 23 fish species were captured from each site, and species richness explained 64–70% of the variance in IBI scores. Fish collections were dominated by insectivores tolerant of habitat and water quality degradation. Suckers and piscivores were relatively uncommon.

The IBI scores were generally not reflective of physical habitat conditions. Variation in IBI scores was indicative of only the grossest differences in physical habitat quality. Weak relationships between physical habitat quality and IBI scores may have been due to large temporal variations in biotic integrity typical of degraded habitats. Alternatively, water quality degradation, which we did not measure, may have confounded relationships between physical habitat and fish metrics.

Regional application of the IBI as a habitat assessment tool in landscapes with widespread physical degradation must overcome lack of suitable reference sites, large temporal variation in IBI scores, and small numbers of fish per collection, leading to lower confidence levels for IBI scores. The scarcity of lightly impacted sites may hinder detection of biotic integrity response along gradients of physical habitat quality.

Introduction

Despite the continued emphasis by resource agencies on toxicity testing and monitoring physical and chemical water quality, a more broad-based approach for defining stream quality is gaining acceptance (Karr, 1991, 1992, and 1993). The index of biotic integrity (IBI), with slight to moderate modifications of its original formulation (Karr *et al.*, 1986), has been used to characterize streams experiencing various types of impacts (nonpoint source pollution, point source pollution, channelization, gravel mining, etc.) in Illinois (Angermeier & Schlosser, 1987), West Virginia (Leonard & Orth, 1986), Missouri (Berkman *et al.*, 1986), Ontario (Steedman, 1988), Louisiana (Killgore & Douglas, 1988), Idaho (Bennett & Fisher, 1989), Oregon (Hughes & Gammon, 1987), Wisconsin (Kanehl & Lyons, 1992), Tennessee (Crumby *et al.*, 1990), Ohio (Rankin & Yoder, 1990), Colorado (Bramblett & Fausch, 1991), and elsewhere (Miller *et al.*, 1988; Fausch *et al.*, 1984). Indices of biotic integrity offer many advantages over other statistics commonly used to describe stream fish communities (Fausch *et al.*, 1990) and to assess environmental quality of watersheds (Karr, 1987). Furthermore, extension of the concepts underlying the IBI to lakes, reservoirs,

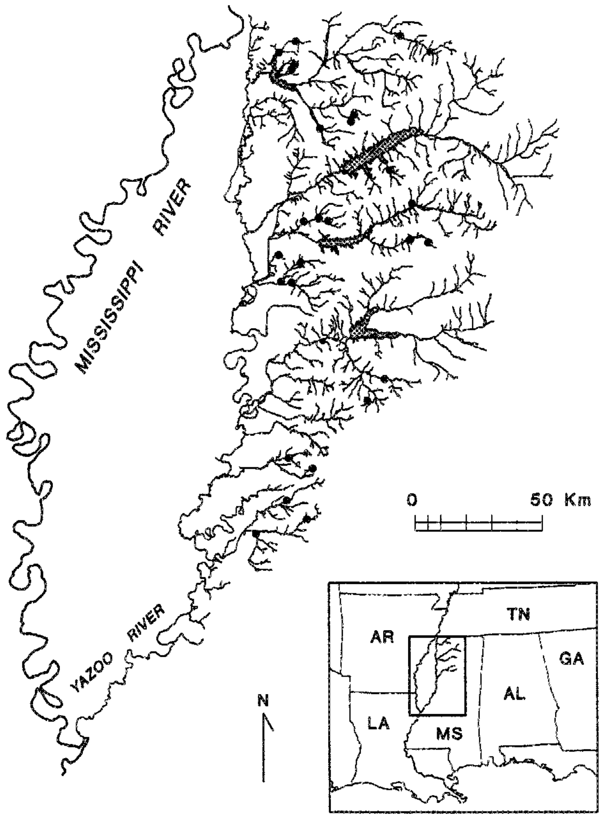

Fig. 1 Study site locations.

Table 1. Numbers of fish species reported in recent studies of rapidly eroding streams of northwest Mississippi

Maximum drainage area, km^2	Number of sites	Duration of study	Total number of fish species	Reference
133	8	Summer, 1985	39	Cooper & Knight (1987)
91	11	1986–1988	31	Knight & Cooper (1990)
253	56	1985	52	Knight & Cooper (1987)
≈500	8	1989–1991	37	Cooper *et al.* (1992)
550	12	1986–1988	52	Knight & Cooper (1991b)
205	4	1991	34	Shields *et al.* (1994)

wetlands, and coastal environments is underway (Karr, 1992). Similar approaches have been used to formulate and test indices based on benthic macroinvertebrate data (Karr & Kerans, 1992) and based on combinations of metrics describing benthic macroinvertebrates, amphibians, and fish collections (Bennett & Fisher, 1989).

In order to verify its validity, the IBI has been shown to correlate with measures of habitat quality and environmental impact (Karr *et al.*, 1984; Leonard & Orth, 1986; Berkman *et al.*, 1986, Lyons 1992). In at least one case, a habitat index was shown to correlate with IBI in order to verify its utility (Petersen, 1992). The State of Illinois has developed a regression equation to predict the 'potential' IBI in terms of habitat variables (Joseph, 1993). Herein we present physical habitat measurements and indices of biotic integrity for warmwater streams along the bluffline bordering the lower Mississippi river alluvial plain in northwestern Mississippi. In this region, a combination of geological and cultural factors has led to large-scale watershed instability and channel erosion resulting in several meters of channel bed lowering and increases in channel width of up to 1000% (Whitten & Patrick, 1981; Simon & Robbins, 1987; Grissinger & Murphey, 1986). We reasoned that physical habitat degradation in our study region would be reflected in fish communities (Shields *et al.*, 1994, Berkman & Rabeni, 1987; Frissell *et al.*, 1986) and thus in IBI scores. The overall objective of this study was to identify relationships between habitat conditions and fish assemblages across a region impacted by channel incision. If successful, these relationships would contribute to developing fish habitat restoration strategies. Since channel incision threatens stream corridor habitats in many areas (Emerson, 1971; Piest *et al.*, 1977; Hortle & Lake; 1982; Galay, 1983; Simon, 1989; Patrick *et al.*, 1991; Kesel & Yodis, 1992), workers in regions outside our own may find this information useful.

Study areas

For this study, we selected 27 stream reaches in hilly watersheds with incised channel networks along the bluffline which borders the alluvial plain of the Mississippi River in northwestern Mississippi (Fig. 1). Contributing drainage areas ranged from 6.2 to 264 km^2, while stream orders ranged from 1 to 4. Following European settlement, these watersheds experienced deforestation, extensive gullying of hillslopes and valley sedimentation (up to 3 m), and more recently (within the last ~40 yr), channel straightening and rapid channel erosion. Reported suspended sediment yield was about 1100 t km^{-2} yr^{-1} (Grissinger *et al.*, 1991; Rebich, 1993). Higher order channels tended to be

Table 2. Trophic classifications of fish species captured from 27 northwest Mississippi sites

Common	Family	Genus	Species
		Omnivores	
River Carpsucker	Catostomidae	*Carpiodes*	*carpio*
Smallmouth Buffalo		*Ictiobus*	*bubalus*
Black Buffalo		*Ictiobus*	*niger*
Gizzard Shad	Clupeidae	*Dorosoma*	*cepedianum*
Threadfin Shad		*Dorosoma*	*petenense*
Common Carp	Cyprinidae	*Cyprinus*	*carpio*
Red Shiner		*Cyprinella*	*lutrensis*
Bullhead Minnow		*Pimephales*	*vigilax*
Striped Shiner		*Luxilus*	*chrysocephalus*
Golden Shiner		*Notemigonus*	*crysoleucas*
Bluntnose Minnow		*Pimephales*	*notatus*
Channel Catfish	Ictaluridae	*Ictalurus*	*punctatus*
Freshwater Drum	Sciaenidae	*Aplodinotus*	*grunniens*
		Insectivorous Cyprinids	
Blacktail Shiner	Cyprinidae	*Cyprinella*	*venusta*
Pugnose Minnow		*Opsopoeodus*	*emiliae*
Emerald Shiner		*Notropis*	*atherinoides*
Creek Chub		*Semotilus*	*atromaculatus*
Redfin Shiner		*Lythrurus*	*umbratilis*
Taillight Shiner		*Notropis*	*maculatus*
Bluntface Shiner		*Cyprinella*	*camura*
Yazoo Shiner		*Notropis*	*rafinesquei*
		Piscivores	
Largemouth Bass	Centrarchidae	*Micropterus*	*salmoides*
Spotted Bass		*Micropterus*	*punctulatus*
Flathead Catfish	Ictaluridae	*Pylodictis*	*olivaris*
Shortnose Gar	Lepisosteidae	*Lepisosteus*	*platostomus*
Longnose Gar		*Lepisosteus*	*osseus*
Spotted Gar		*Lepisosteus*	*oculatus*
Chain Pickerel	Esocidae	*Esox*	*niger*

wide and flanked by large bars or berms, while low order channels were narrow and often incised into hard clay (Harvey & Watson, 1986; Simon, 1989). Streams were typically lacking in pool habitat, were dominated by sandy substrates, and had low woody debris densities (Shields *et al.*, 1994). Available water quality data indicated acceptable conditions for most warmwater species (Slack, 1992; Cooper *et al.*, 1992; Cooper & Knight, 1991). Land use was generally non-urban, and cultivation was generally restricted to valley bottoms and comprised only 20–30% of watershed area. Hillslopes were pasture or forest.

Little is known about the aquatic fauna of this region prior European settlement, which occurred *c.* 1835 and was followed by rapid deforestation, hillslope erosion, and valley sedimentation (Hilgard, 1860). Ross & Brenneman (1991) listed 123 species found currently or historically in the Yazoo River basin.

Table 2. (Continued)

Common	Family	Genus	Species
		Non-cyprinid insectivores	
Spotted Sucker	Catostomidae	*Minytrema*	*melanops*
Pirate Perch	Aphredoderus	*Aphredoderus*	*sayanus*
Brook Silversides	Atherinidae	*Labidesthes*	*sicculus*
Blacktail Redhorse	Catostomidae	*Moxostoma*	*poecilurum*
Creek Chubsucker		*Erimyzon*	*oblongus*
Lake Chubsucker		*Erimyzon*	*sucetta*
White Crappie	Centrarchidae	*Pomoxis*	*annularis*
Bluegill		*Lepomis*	*macrochirus*
Redear Sunfish		*Lepomis*	*microlophus*
Warmouth		*Lepomis*	*gulosus*
Longear Sunfish		*Lepomis*	*megalotis*
Green Sunfish		*Lepomis*	*cyanellus*
Spotted Sunfish		*Lepomis*	*punctatus*
Dollar Sunfish		*Lepomis*	*marginatus*
Banded Pygmy Sunfish	Elassomatidae	*Elassoma*	*zonatum*
Blackstriped Topminnow	Fundulidae	*Fundulus*	*notatus*
Blackspotted Topminnow		*Fundulus*	*olivaceus*
Golden Topminnow		*Fundulus*	*chrysotus*
Yellow Bullhead	Ictaluridae	*Ameiurus*	*natalis*
Brown Bullhead		*Ameiurus*	*nebulosus*
Black Bullhead		*Ameiurus*	*melas*
Brindled Madtom		*Noturus*	*miurus*
Brown Madtom		*Noturus*	*phaeus*
Johnny Darter	Percidae	*Etheostoma*	*nigrum*
Dusky Darter		*Percina*	*sciera*
Brighteye Darter		*Etheostoma*	*lynceum*
Blacksided Darter		*Percina*	*maculata*
Harlequin Darter		*Etheostoma*	*histrio*
Redfin Darter		*Etheostoma*	*whipplei*
Gulf Darter		*Etheostoma*	*swaini*
Goldstripe Darter		*Etheostoma*	*parvipinne*
Yazoo Darter		*Etheostoma*	*raneyi*
Mosquito Fish	Poeciliidae	*Gambusia*	*affinis*
		Herbivores	
Central Stoneroller	Cyprinidae	*Campostoma*	*anomalum*
Miss Silvery Minnow		*Hybognathus*	*nuchalis*
		Detritivore	
Southern Brook Lamprey	Petromyzontidae	*Ichtyomyzon*	*gagei*
		Parasite	
Chestnut Lamprey	Petromyzontidae	*Icthyomyzon*	*castaneus*

Recent studies of streams within this study area have captured 31 to 52 fish species (Table 1), although many occurred in fragmented ranges.

Data collection

At each study site, we sampled 200–400 m of channel for fishes and physical habitat variables during the spring of 1992 and summer and early fall of 1993. Acquisition of fish samples and physical data general-

Table 3. Intolerant and tolerant fish species

Common	Family	Genus	Species
		Intolerant species	
Spotted Sucker	Catostomidae	*Minytrema*	*melanops*
Blacktail Redhorse		*Moxostoma*	*poecilurum*
Central Stoneroller	Cyprinidae	*Campostoma*	*anomalum*
Taillight Shiner		*Notropis*	*maculatus*
Striped Shiner		*Luxilus*	*chrysocephalus*
Brown Madtom	Ictaluridae	*Noturus*	*phaeus*
Blackside Darter	Percidae	*Percina*	*maculata*
Harlequin Darter		*Etheostoma*	*histrio*
Gulf Darter		*Etheostoma*	*swaini*
Yazoo Darter		*Etheostoma*	sp
Chain Picker	Esocidae	*Esox*	*niger*
		Tolerant species	
River Carpsucker	Catostomidae	*Carpiodes*	*carpio*
Green Sunfish	Centrarchidae	*Lepomis*	*cyanellus*
Common Carp	Cyprinidae	*Cyprinus*	*carpio*
Red Shiner		*Cyprinella*	*lutrensis*
Golden Shiner		*Notemigonus*	*crysoleucas*
Bluntface Shiner		*Cyprinella*	*camura*
Yazoo Shiner		*Notropis*	*rafinesquei*

ly required two to three hours of work by 6 persons. Sampled reach lengths were equivalent to 17 to 151 water surface widths (mean = 42, s.d. = 28). The temporal and spatial frequency of channel features such as pools, runs and riffles was erratic due to the highly disrupted nature of stream morphology, but in all cases, reach lengths were long enough to accurately reflect site quality (Angermeier & Karr, 1986).

Fish were collected from each site at base flow using a Coffelt BP-4 backpack mounted electroshocker. Sampled reaches were fished for a total of 421 to 2819 sec (mean = 1176, s.d. = 471) of electric field application by a three-person crew wading upstream. One person carried the electroshocker, while two captured stunned fishes in nets. The crew took care to thoroughly sample banklines and habitat structures not on the bank-riffles, snags, holes, etc. Effort was applied at each structure until catch per unit of effort began to decline. If no fish were produced in response to initial effort, the crew continued wading upstream.

Fishes longer than about 15 cm were identified, measured for total length, and released. Weights were estimated using length-weight regression formulas developed from previous collections of fish from northern Mississippi. Smaller fish, and larger fish that could not be identified in the field, were preserved and transported to the laboratory for identification and measurement of weight and length. Fish shorter than 2 cm were not used in IBI computations to avoid artificially inflating species richness by including young of the year of rare species (Fausch *et al.*, 1990). In both the field and laboratory, presence of lesions or other anomalies on fishes were noted during processing.

Water pH, conductance, dissolved oxygen, temperature were measured immediately before or after electroshocking using a Martek water quality meter. Physical aquatic habitat variables were sampled from each study reach concurrent with fish sampling. The design of physical data collection and analysis procedures was partially based on work by others (e.g. Gorman & Karr, 1978; Petersen, 1992; Plafkin *et al.*, 1989; Berkman *et al.*, 1986), but was intended to describe conditions in the channelized, eroding channels we studied. We sampled depth and bed type at 100 points at each site arrayed at roughly equal intervals along 20 transects. Side channels and embayments were sampled only if they were hydraulically connected to the main channel at up and downstream ends. Depth was measured with a wading rod, and bed type was visually classified as clay, sand, gravel, riprap[1], vegetation, debris, or other (e.g., man-made items). Measured water surface width and visual estimates of top bank width, channel depth, and bank vegetation were also recorded for each of the 20 transects. The presence and number of beaver dams and manmade structures (e.g., weirs, revetments[2], jacks[3], etc.) within or immediately downstream from each reach were noted, and the area of each large woody debris formation in the plane of the water surface was visually estimated. Contributing drainage areas and stream orders were computed using U.S. Geological Survey 1:100 000 scale topographic maps. Ephemeral channels were included in drainage network analysis for stream order determination. Channel bed slope measurements and bed sediment size distributions were obtained for as many sites as possible from literature and data files of the National Sedimentation Laboratory.

[1] Angular pieces of quarried limestone ~ 10 to 40 cm in diameter placed for erosion control.

[2] Blankets of riprap placed on eroding banks.

[3] Erosion control devices made by joining three 5-m long concrete beams at their midpoints Jacks are usually placed in arrays along eroding banks and fastened together with steel cables.

Data analyses

We computed species composition for each fish collection. Each species collected was assigned to an appropriate trophic guild using our own expertise backed up with reference to work done by others (e.g., Karr *et al.*, 1986, Angermeier & Schlosser, 1987; Saylor & Ahlstedt, 1990; Robison & Buchanan, 1988) (Table 2). When trophic status for younger fishes differed from larger, mature individuals, we used the latter.

As shown in Table 3, we classified 13 of the 65 species on our master list as intolerant and 7 as tolerant based on our previous observations (Table 1) and work by Jester *et al.* (1992). Tolerants included the green sunfish (*Lepomis cyanellus*), which is often regarded as tolerant of environmental degradation (Karr *et al.*, 1986), and two species that specialize in shallow, sandy stream habitats, the bluntface shiner (*Cyprinella camura*) and the Yazoo shiner (*Notropis rafinesquei*) (Suttkus, 1991). These shiners were ubiquitous in the degraded reaches of our study area because of the prevalence of shallow, sandy habitat in the eroded channels. Although they would be regarded as habitat specialists and perhaps as intolerant in some regions (e.g., Jester *et al.* (1992) rated *Cyprinella camura* as intolerant of both water quality and habitat degradation), we used them as indicators of the level of disturbance due to channel erosion.

In order to compute IBI scores, we had to modify the original formulation (Karr *et al.*, 1986) to allow for regional faunal differences and the scarcity of unimpacted or lightly impacted streams for reference sites. Metrics used to formulate the original IBI were modified in three ways consistent with recommendations by Karr *et al.* (1986) for applications outside the midwestern U.S. First, the metric proportion of individuals as green sunfish was replaced by proportion of individuals that are members of tolerant species. Although green sunfish tended to dominate many of the collections, other sites were populated primarily by one or more of the tolerant species listed in Table 2. Therefore the metric was broadened. Secondly, *Micropterus* spp. were not excluded when enumerating sunfish species because our experience indicates that these species are particularly sensitive to the kinds of habitat degradation common to our study region. Thirdly, since electrofishing times varied from site to site, catch per unit of effort (measured in units of fish per minute) was used as the metric of fish abundance instead of the number of fish in each sample.

Due to a lack of suitable reference sites, scoring criteria for each IBI metric were set at natural breakpoints in the data or, if distributions were not clustered, at levels so that the data set was divided roughly into thirds. This approach led to development of an index of relative biotic integrity, and the results should be interpreted accordingly. Plots of species richness metrics against upstream watershed area showed no relationships (Fig. 2), and therefore scoring criteria were not adjusted for stream size. Metrics and scoring criteria are listed in Table 4 along with values proposed by Karr *et al.* (1986).

Physical habitat data were used to compute 13 metrics of physical habitat quality (Table 5). Selection of metrics was based upon earlier work elsewhere (Plafkin *et al.*, 1989, Osborne *et al.*, 1991), in this region (Shields *et al.*, 1994), and similar work detailing physical habitat/fish community relationships in disturbed warmwater streams, and was intended to track physical (but not chemical, i.e., water quality) degradation likely to impact biotic integrity. Five metrics reflected riparian conditions: the proportion of water surface directly underneath shade canopy and the proportion of bank line supporting large trees, smaller woody vegetation, herbaceous vegetation, and the exotic vine, kudzu (*Pueraria lobata*). Kudzu suppresses native riparian woody species in our study region (Shields *et al.*, 1993), and offers and offers virtually no shade or bank stabilization benefit.

Two metrics were related to the severity of channel incision and attendant accelerated channel erosion: channel (not water) depth and top width. Two metrics, the proportion of bed surface as sand and as gravel, directly reflected substrate conditions. Sediment coarser than gravel was virtually absent in our study region. We combined substrate and depth to generate a metric of habitat richness (Table 5). Since manmade structures often furnish the only pool habitat and cover in the disturbed streams studied, we formulated a metric to express structural influence based on estimates of the relative area of hydraulic influence of each structural type (Table 5). Pool habitat availability was the final physical metric.

The utility of the relative IBI for quantifying effects of various physical habitat deficiencies on fish was tested by computing correlation coefficients between biotic and physical metrics. In addition, distributions of physical metrics for sites with high and low relative biotic integrity were compared using a nonparametric (Mann-Whitney U) test.

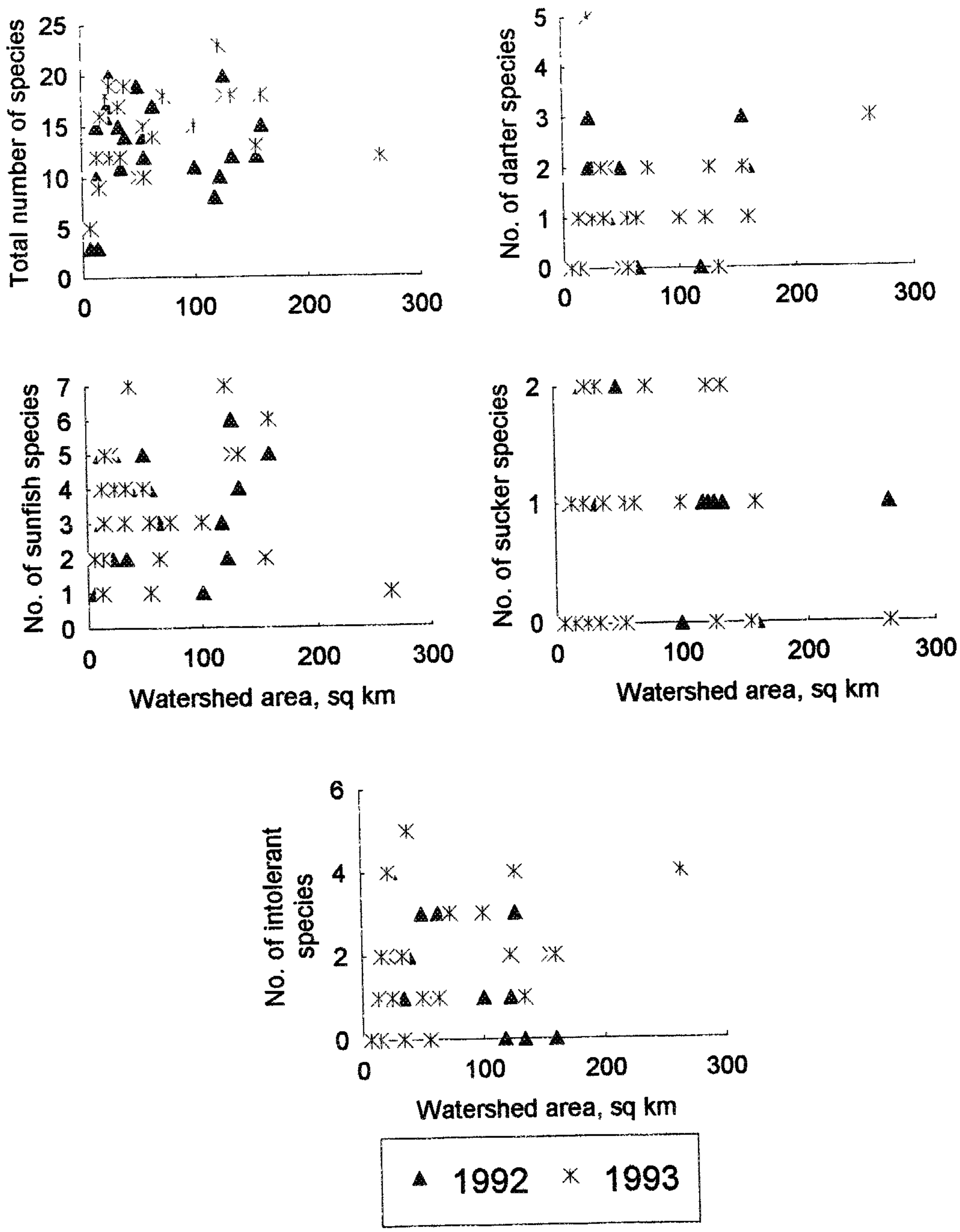

Fig 2. Species richness metrics versus contributing watershed area.

Results

A total of 27 sites were sampled. Twenty-six sites were sampled in 1992; and 24 of these were sampled in 1993, plus one additional site, giving a total of 51 data sets. Fish sampling yielded a total of 16 707 individuals representing 65 species and 2 hybrids. Individual collections ranged from 24 to 1475 individuals (mean = 326, median = 222) representing 3 to 23 species per site per sampling date (a 'collection'). Mean numerical catch was 23% lower for spring and early summer (1992) than summer and fall (1993) samples, but variances were homogeneous, and the difference in annual means was not statistically significant (p = 0.20, paired two-

Table 4. Scoring criteria used to compute indices of relative biotic integrity from fish collections from degraded northwest Mississippi streams Criteria proposed by Karr *et al* (1986) for midwestern streams are shown in parentheses

Category	Metric	Scores 1	3	5
Species richness and composition[2]	Total number of species	0–9 (0–3)	10–16 (4–9)	≥ 17 (≥10)
	Number of darter species	0 (0)	1 (1-2)	≥2 (≥3)
	Number of sunfish species[3]	0–2 (0)	3–4 (1)	≥5 (≥2)
	Number of sucker species	0 (0)	1 (1)	≥2 (≥2)
	Number of intolerant species	0 (0)	1–2 (1–2)	≥3 (≥3)
Trophic composition	Proportion of individuals that are members of species classified as tolerant[4]	≥35% (> 20%)	20% ≤ x < 35% (5%–20%)	≤ 20% (< 5%)
	Proportion of individuals as omnivores	≥20% (> 45%)	5% ≤ x < 20% (20%–45%)	< 5% (< 20%)
	Proportion of individuals as insectivorous cyprinids	< 10% (< 20%)	10% ≤ x < 50% (20%–45%)	≥ 50% (> 45%)
Fish abundance and condition	Proportion of individuals as piscivores	< 1% (< 1%)	1% ≤ x < 2% (1%–5%)	≥ 2% (>5%)
	CPUE, fish per minute[5]	< 8	8 ≤ x < 20	≥ 20
	Proportion of individuals as hybrids	≥ 2% (>1%)	0.10% ≤ x < 2% (>0%–1%)	< 0.10% (0%)
	Proportion of individuals with disease or anomaly	≥ 2% (> 5%)	1% ≤ x < 2% (>2–5%)	<1% (0–2%)

[2] Scoring criteria for these metrics vary with stream size and region. Those shown in parentheses are for third-order sites on the Embarras River, Illinois (Fausch *et al*, 1984 in Karr *et al.*, 1986) and are presented here for general comparison only.
[3] Our metric included *Micropterus* spp., while Karr *et al.* (1986) did not. See text for details.
[4] Metric proposed by Karr *et al.* (1986) was proportion of individuals as green sunfish. See text for discussion.
[5] Scoring criteria given by Karr *et al.* (1986) were in units of total individuals in each sample.

sample *t* test). Twelve of the 51 collections contained fewer than 100 fish, and 39 contained fewer than 400. Fishes longer than 20 cm were extremely rare, and were absent at shallower sites.

Only 4 species comprised 52% of the numerical catch: the mosquito fish, *Gambusia affinis* (8%), and the cyprinids *Cyprinella camura* (20%), *Notropis rafinesquei* (15%), and *Semotilus atromaculatus* (9%). *Cyprinella camura* appeared in 38 of the 51 collections. Five of the 65 species captured and one of the hybrids were represented by only one individual, and 20 species were represented by 9 or fewer fish. Tolerant species comprised as much as 80% of individual collections and averaged 35%. Four of the 13 most abundant species, including the two most abundant were ones we rated as tolerant (Table 2). Furthermore, 7 of the 11 most abundant species, comprising 35% of the total numerical catch, were rated as tolerant or moderately tolerant of water quality and habitat degradation by Jester *et al.* (1992). Hybrids appeared in only 5 of the 51 collections, but may have been underestimated due to difficulties in identification (Killgore & Douglas, 1988). Fish with anomalies, parasites, and diseases were found in 26 of the 51 collections. Sunfishes were present in all collections and averaged 3 species per collection. Suckers were less common, averaging only

Table 5 Metrics used to characterize physical habitat quality

Category	Metric	How computed	Units
Riparian conditions	Proportions of bank line supporting large (> 5 m tall) trees, smaller woody vegetation, grasses and other herbaceous vegetation, and exotic vine, Pueraria lobata (kudzu)	Visual estimate of proportion of bank covered with each of the 4 vegetation types	Percent
	Canopy	Average of visual estimate at each of 20 transects	Percent
Severity of channel incision	Channel depth	Channel depth based on visual estimate at each of 20 transects, water depth was average of measurements at 100 grid points	m/m
	Top width of channel	Channel width was average of visual estimate at each of 20 transects, water width was average of measured values at each transect	m/m
Substrate and habitat heterogeneity	Proportion of bed surface as sand	Visual classification of bed type made at each of 100 grid points, number sand/number gravel	Percent
	Proportion of bed surface as gravel	Visual classification of bed type made at each of 100 grid points; number sand/number debris	Percent
	Habitat richness	The number of 'habitat types' represent in the grid point data were enumerated. Four depth categories (0- and 6 substrates were recognized for a total of 24 possible habitat types. Depth categories were chosen to be representative of distinct fish habitats based on experience gained from previous ichthyofaunal surveys of watersheds in the region.	Count
Cover and pool formation	Large woody debris density	Sum of visually estimated areas of all LWD formations in the plane of the water surface divided by water surface area	m^2/km^2
	Influence of structures	Proportion of reach influenced by structure $= [\sum$ length of zones of influence$]^1$ /reach length in m	m/m
	Availability of pool habitat	Proportion of water depth measurements made at each of 100 grid points concurrent with fish sampling > 30 cm	Percent

[1] Σ length of zones of influence = { [25 m × number of spur dikes] + [length of riprap revetment in m] + [1 m × number of Kellner jacks]}/ 2 + [100 m × number of grade control structures] + [10 m × number of beaver dams]

Weights for structure types were assigned based on field observations of typical regions of hydraulic influence. Note that zone lengths for structures placed on bank are divided by 2

1 species per collection and were absent in 19 of the 51 collections. No collection contained more than 2 sucker species. Darters and intolerants were present in 37 and 39 collections, respectively. Up to 5 intolerant species were found in each collection, with an average of 2 species per collection. Trophic composition of fish collections averaged 15% omnivores, 38% insectivorous cyprinids, and 3% piscivores. No piscovores were found in 15 collections, and piscivores comprised 1% or less of 29 of the 51 collections. Collections from shallower sites were dominated by cyprinids and centrarchids, while centrarchids were more numerous at deeper sites. Suckers were relatively rare at shallow sites, while darters were more common at shallower sites than deeper ones.

IBI scores based on the 51 collections ranged from 24 to 50, which compares with a possible range of 12 to 60. Mean IBI scores were not significantly different between years (paired, two-sample t-test, $p = 0.27$). Site by site comparison of 1992 and 1993 IBI scores revealed variations ranging from −14 to +16 (Fig. 3), and the absolute value of these variations was not correlated with either 1992 or 1993 IBI score. Distributions of IBI scores and component metrics were similar for the two years, but the 1993 data displayed slightly higher levels of species richness, and IBI scores were accordingly skewed towards higher values (Fig. 4).

Table 6 Correlation coefficients between biotic and physical metrics with $p<0.05$ Entries in standard font are for 1992 data, and italicized entries are for 1993 data

Metric	Channel depth	Channel width	Proportion of bed surface as gravel	Structural influence	Canopy	Grass or herbaceous vegetation	Kudzu	Woody vegetation smaller than 5 m high	Trees > 5 m high	Habitat richness	Water width	Availability of pool habitat
Total no. of species				*0.43*								
No. of darter species	*−0.38*	*−0.36*	*−0.38*		0.47 *0.46*		*−0.39*	*0.40*				
No. of sucker species		*0.35*		*0.52*								
No. of intolerant species	−0.37 *−0.35*		*−0.38*		0.35	0.53	−0.38 *−0.40*					*0.51*
Proportion of individuals classed as tolerant species	*0.34*						0.51			*0.37*	−0.42	−0.48
Proportion of individuals classed as omnivores				0.45			−0.46				0.39	
Proportion of individuals classed as insectivorous cyprinids							0.43					
Proportion of individuals classed as piscivores				0.42								0.48
Catch per unit of effort		0.35				−0.35	0.38, *0.53*		−0.34, *−0.34*		−0.42	−0.36, *−0.38*

The total number of species explained 70% and 64% of the variance in IBI scores in 1992 and 1993, respectively.

In order to assess the temporal variation in IBI-based stream classification, each of the two annual data sets were bisected about their midpoints: sites with IBI scores >37 were designated as sites with high relative biotic integrity, while those with scores <37 were in the low relative biotic integrity category. Of the 24 sites which were sampled during both years, 16 remained in the same category both years.

Water quality observations were consistent with findings of more extensive studies (Slack, 1992; Cooper, McCoy, & Knight, 1992; Cooper & Knight, 1991;

Table 7 Comparison of physical habitat metrics and water quality for sites with low (<37) and high (>37) relative biotic integrity[7]

	1992			1993		
	Mean for 15 sites with IBI >37	Mean for 9 sites with IBI <37	Mann-Whitney [8] probability	Mean for 11 sites with IBI >37	Mean for 13 sites with IBI <37	Mann-Whitney[7] probability
Watershed area, km^2	76	56	0.929	59	74	0 149
Channel depth, m	5	4	0.200	4	4	0 463
Channel top width, m	25	22	0 531	20	21	0 605
Proportion of bed surface as sand	63%	64%	0 881	63%	54%	0 644
Proportion of bed surface as gravel	23%	22%	0 880	15%	25%	0.589
Structural influence	9%	17%	0.287	3%	15%	0.076
Woody debris density (m^2/km^2)	15,354	18,860	0 194	39,289	53,110	0.264
Canopy (%)	32%	24%	0.371	46%	38%	0 549
Grass	14%	17%	0 591	19%	20%	0 479
Kudzu	7%	3%	0 238	18%	9%	0 286
Brush < 5 m high	31%	28%	0 531	29%	33%	0.369
Trees >5 m high	10%	6%	0 387	10%	9%	0.935
Water width (m)	7	7	0.788	6	7	0.430
Habitat richness	8	10	0.126	9	10	0.281
Availability of pool habitat	18%	21%	0.282	19%	21%	0.496
Temperature, deg C	23 1	24 5	0.405	26.8	26.0	0 860
Specific conductivity, μmhos cm^{-1}	235	83	0.057	281	116	0 972
Dissolved oxygen, mg L^{-1}	8.1	7 8	0 942	7 5	8 0	0 307
pH	7.0	6 6	0.469	7.2	7.4	0 503

[7] Sites with IBI scores = 38 were excluded from this analysis to insure a minimum difference of 4 between IBI scores of low and high classifications.

[8] The value shown represents the probability that the difference between the ranks of the measurments could arise due to chance alone

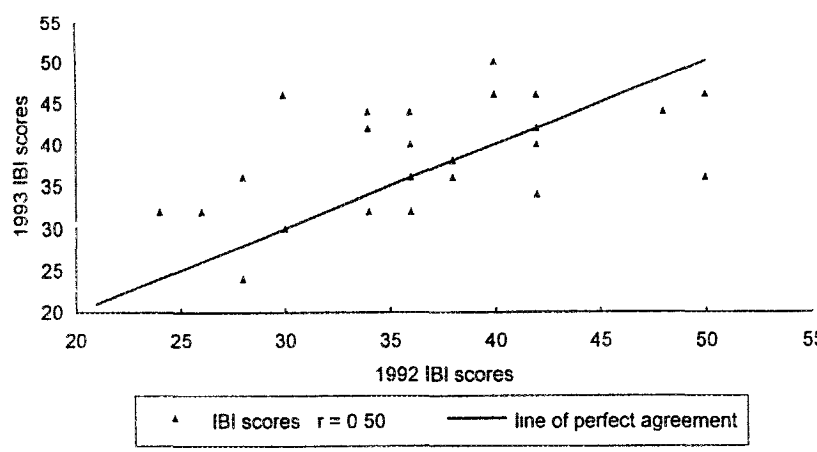

Fig. 3. Relative index of biotic integrity based on collections made in late spring and early summer, 1992 versus relative index of biotic integrity based on collections made at the same sites during summer and fall, 1993.

Table 8. Comparison of biotic metrics for sites with low (<37) and high (>37) relative biotic integrity[9]

	1992			1993		
	Mean for 15 sites with IBI > 37	Mean for 9 sites with IBI < 37	Mann-Whitney probability[10]	Mean for 11 sites with IBI > 37	Mean for 13 sites with IBI < 37	Mann-Whitney probability[10]
Total number of species	**10**	**17**	0.001	**11**	**17**	<0.001
Number of darter species	1	1	0.110	1	2	0.070
Number of sunfish species	**3**	**4**	0.058	**3**	**4**	0.014
Number of sucker species	**0**	**1**	0.002	**0**	**1**	<0.001
Number of species classified as intolerant	**1**	**3**	0.002	**1**	**2**	0.009
Percent individuals as tolerants	42%	31%	0.222	33%	28%	0.341
Percent individuals as omnivores	15%	13%	0.976	18%	13%	0.644
Percent individuals as insectivorous cyprinids	31%	39%	0.297	46%	33%	0.369
Percent individuals as piscivores	1%	5%	0.051	**1%**	**4%**	0.031
Catch per unit of effort (fish/min)	11	17	0.571	19	15	0.807
Percent individuals as hybrids	0.30%	0.00%		0.02%	0.07%	
Percent individuals as with lesions, parasites, or body anomalies	2.35%	0.27%		1.17%	0.58%	0.536
IBI	**31**	**44**	<0.001	**33**	**44**	<0.001

[9] Sites with IBI scores = 38 were excluded from this analysis to insure a minimum difference of 4 between IBI scores of low and high classifications. Boldface indicates $p < 0.05$.
[10] The value shown represents the probability that the difference between the ranks of the measurments could arise due to chance alone

Cooper & Knight, 1987). Measured values of temperature ranged from 12.6 to 32.3 °C, dissolved oxygen from 5.1 to 13.8 mg l^{-1}, and pH from 5.6 to 8.8, indicating acceptable conditions for aquatic life. Specific conductance ranged from 14 to 1178 μmhos cm^{-1}. Three values of specific conductance exceeded 900 μmhos cm^{-1}, likely indicating undesirably high levels of dissolved ions.

Physical habitat conditions were typical of incised channels in this region (Shields & Hoover, 1991). Effects of channel straightening and accelerated channel erosion were manifest by deep channel incision and raw, eroding banks or, less commonly, by high sediment loads, floodplain sand deposits, and recent conversion of bottomland to wetland habitat in lower reaches of the incising watersheds. Channel widths and depths averaged 23 m and 4 m, respectively, but mean

Table 9. Physical habitat and water quality characteristics for sites with maximum and minimum relative biotic integrity scores

Physical Habitat Metric	1992		1993	
	Two sites with lowest relative IBI	Two sites with highest relative IBI	Two sites with lowest relative IBI	Two sites with highest relative IBI
Proportion of bank line supporting woody vegetation, percent	26–28	24–49	25–39	24–49
Canopy, percent	0–1	4–7	31–53	4–7
Channel depth, (m)	4–5	4–4	6–6	4–4
Top width of channel, (m)	16–22	32–40	15–24	32–40
Proportion of bed surface covered with sand, percent	16–52	40–46	35–40	40–46
Proportion of bed surface covered with gravel, percent	48–80	42–46	0–53	42–46
Habitat richness, number of depth category-bed type combinations observed	4–7	10–16	10–15	10–16
Large woody debris density, m^2 km^{-2}	0–1,030	11,900–20,100	0–207	11,930–20,102
Structural influence, percent	0–0	0–28	0–17	0–28
Availability of pool habitat, percent water area with depth > 30 cm	0–02	5–7	2–29	5–7
Temperature, deg C	18–25	30[6]	17–27	29–31
Specific conductivity, μmhos cm	120–520	62[6]	82–948	68–166
Dissolved oxygen, mg L^{-1}	6 8–9.8	6 2–7.1	7 0–7 3	7.5–8 5
pH	6.6–8 8	6 4–7.6	6 3–7.5	7 6–7.9
Evidence of other stresses	One of these sites was dry in 1993	None	Numerous dead fish observed at one site	None
Index of relative biotic integrity	24–24	50–50	24–30	46–50

[6] Data were available for only one of the two sites.

water surface widths and depths were only 7 m and 20 cm. Width of incised channels increased with watershed area, but channel depth did not, because the smaller streams were often just as deeply incised as larger ones. Water depths tended to be shallow, and only 20% of the sampled depths exceeded 30 cm. Mean water depth, percent of depths >30 cm, and width were positively correlated with watershed area ($r=0.54$, 0.53, and 0.59, respectively, and $p<0.00004$).

Shade canopy ranged from 0 to 88%, but averaged only 34%. Trees taller than 5 m were recorded for only 9% of the banklines (by length). Woody debris formations occupied areas ranging from 0 to 332 000 m^2 km^{-2} of water surface. Substrates were dominated by sand (60%), with only 11 of the 51 collections having more gravel than sand. At each site, an average of 10 (range 4 to 20) different 'habitat types' (depth-bed type combinations) were recorded. Sediment size distributions were available for 12 sites, and median sizes ranged from 0.27 to 0.60 mm. Bed slopes were available for 11 of the 27 sites, and ranged from 0.8 to 2.0 m km^{-1}. Bed slopes were inversely proportional to watershed size ($r=-0.82$, $p=0.002$), but bed material size was not significantly correlated with slope or watershed size. In 1992, 11 of the 26 sampled sites had some type of bank protection structure, including short stone spurs[4] (4 sites), stone blanket riprap (three sites), a ridge of stone placed along the bank toe (4 sites) and Kellner jacks (one site). By the 1993 sampling, stone spurs and low (0.6 m) stone weirs had each been added to one site each, and a grade control weir had been constructed immediately downstream from another site.

Correlations between physical habitat and IBI metrics that were significantly different from zero at the 95% confidence level are listed in Table 6. None of the correlations were >0.54, and many were unique to only one of the two years. Six pairings were similarly associated in both data sets: the number of darter species was positively correlated with canopy, the number of intolerant species was inversely related to channel depth and kudzu, and numerical catch per unit of effort was positively correlated with kudzu and inversely related to large tree cover and pool habitat availability. These relationships are sensible in light of our qualitative observations. Darters were more common in relatively stable, nonincised reaches with tree-lined banks. Streams with mild or moderate incision tended to have well-vegetated banks and more pool habitat. Incised channels often had treeless banks draped with kudzu. These streams were almost entirely composed of shallow, sandy habitats populated by large numbers of small cyprinids which were easily captured. Riparian vegetation composition has been used as an indicator of stream habitat quality in agricultural watersheds in Missouri (Berkman *et al.*, 1986) and by Petersen (1992) for lowland agricultural watersheds in Sweden, alpine watersheds in Italy and intermontane desert watersheds in Idaho.

Physical conditions found in streams with high and low IBI scores were compared. The midpoint of the observed IBI score range (37) was selected to be the dividing line between high and low relative biotic integrity. Each of the two data sets (1992 and 1993) were divided into subsets representing sites with IBI scores greater than and less than 37. In order to insure that the minimum difference in relative biotic integrity between high and low sites was significant (Karr *et al.*, 1986; Fore *et al.*, in press), sites with IBI scores = 38 were excluded from this analysis. Thus the IBI scores for subsets representing relatively low integrity ranged from 24 to 36, and those representing high integrity ranged from 40 to 50. Means were then computed for selected physical and biological metrics for each group, and

Within our data set, relative biotic integrity was a weak indicator of physical habitat degradation. None of the differences in physical metrics between groups with high relative IBI scores and those with low scores were statistically significant when compared using a MannWhitney U test ($p \geq 0.057$) (Table 7). This was despite the pronounced differences in IBI scores and several of its component metrics (Table 8). The total number of species, number of sunfish, sucker, and intolerant species were significantly different ($p \leq 0.05$) both years.

When sites that represented the extremes in relative biotic integrity were compared, physical differences were much more apparent (Table 9). In 1992, the sites with the top two IBI scores had woody debris densities on the order of 10^4 m^2 km^{-2}, 5–7% of water area with depth>30 cm, and moderate levels of physical richness (10–16 different depth-bed type combinations observed). One of the top two sites was influenced by manmade structure (a ridge of riprap placed along 110 m of the bank toe). In contrast, the two sites with lowest relative biotic integrity in 1992 had no structural influence, were practically devoid of woody debris and pools, and had very low ($S = 4$ to 7) levels of physical habitat richness. Furthermore, one of these two sites was dry when visited in 1993. Water quality differences were not detected, except that high IBI sites tended to have lower specific conductance.

In 1993, one of the top two IBI sites from 1992 repeated as a top site, but the other three sites in this comparison were different from 1992. However, higher levels of relative IBI were still associated with higher woody debris density, manmade structures, pool habitat availability, and lower specific conductance (Table 9). The site with lowest observed relative IBI (24) had a specific conductance of 948 μmhos cm^{-1}.

Discussion

Habitat conditions (shallow depths, sandy substrates, little woody debris or riparian vegetation) and associated fish assemblages (dominated by small individuals of tolerant species) were similar to those reported for other agricultural watersheds in Mississippi and the eastern U.S. that have been impacted by accelerated channel erosion (Shields & Hoover, 1991; Menzel *et al.* 1984). Our failure to define stronger relation-

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[4] Riprap erosion control structures that protrude into the channel at right angles to the flow. Spurs are typically less than 10 m long and are placed on eroding banks at intervals of 15 to 25 m

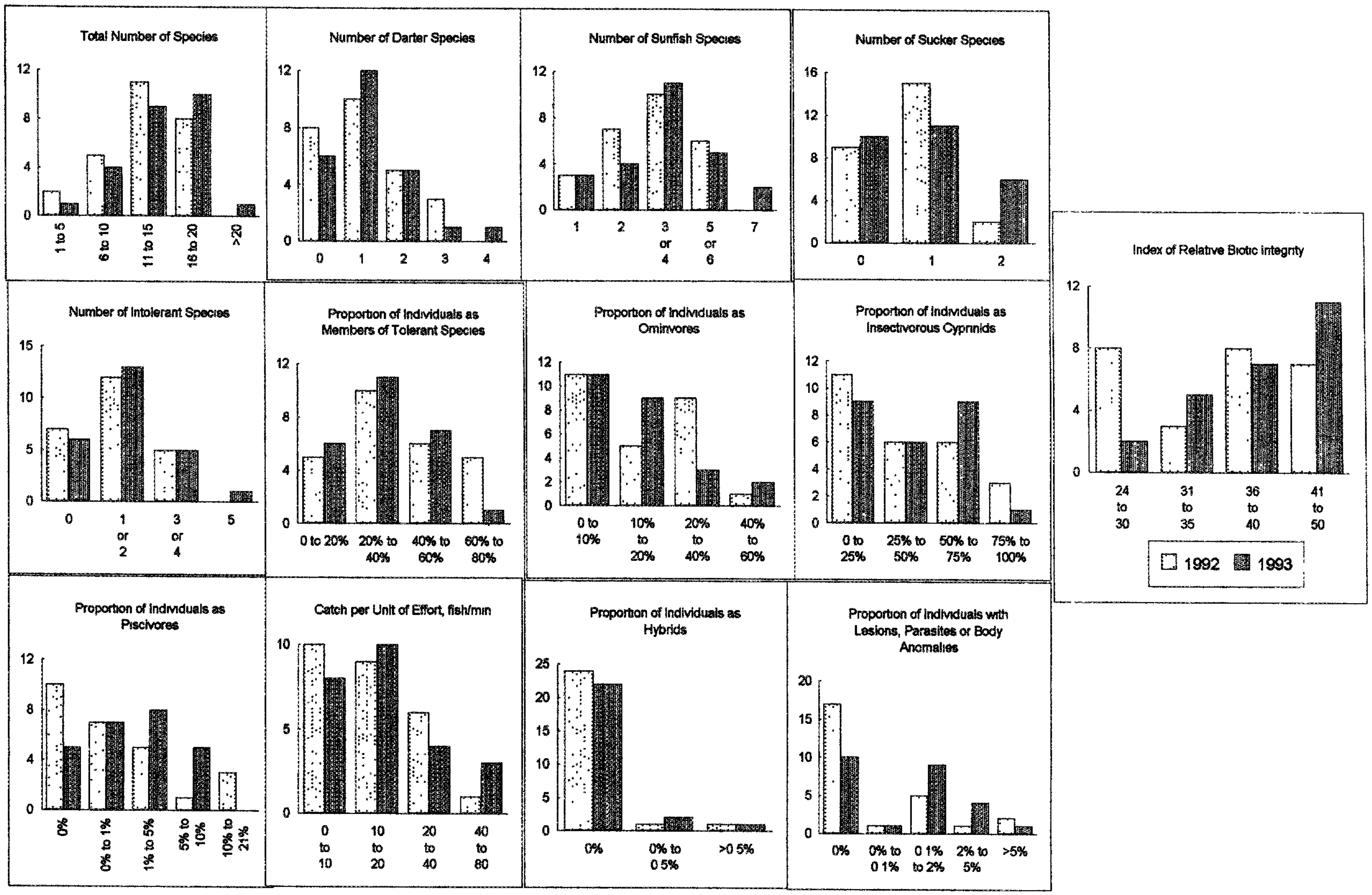

Fig 4 Frequency histograms of IBI metrics and relative IBI scores

ships between physical habitat quality and fish-based indices of relative biotic integrity was disappointing, especially in light of the successes reported by others. Most difficulties were due to the extent of physical stream habitat degradation in our study region. First, and perhaps most important, we were not able to designate unimpacted reference sites. We compensated for this shortcoming by basing our expectations and metric scoring criteria on the data at hand, which led us to computation of indices of relative rather than absolute biotic integrity. Since pristine watersheds are practically nonexistent, all IBI analyses follow a somewhat similar path, but to a lesser degree than in our case. Secondly, IBI scores varied a good bit from one year to the next at a given site. Although some of this variation may have been due to seasonal differences between years, most variation was probably related to habitat quality. Karr *et al.* (1987) found that temporal IBI variation was greater for lower quality sites. In a regional study of Ohio streams, temporal variation of a fish-based IBI was greatest for streams degraded by cultural pollution (Rankin & Yoder, 1990), and the coefficient of variation for IBI was negatively correlated with a habitat quality index. The levels of temporal variation we observed were high relative to variations reported by Karr *et al.* (1987) and Steedman (1988) but were in line with variations reported by Rankin & Yoder (1990) for reaches with degraded habitats. Conversely, western ecosystems may behave differently. Bramblett & Fausch (1991) noted several problems with applying an IBI to a portion of the Arkansas River basin in southeastern Colorado characterized by low habitat diversity, low woody debris density, and highly variable flow regimes. In particular, IBI scores showed a temporal upward trend during a period of impact.

A third problem we experienced was related to low levels of fish abundance. The majority of our fish collections were smaller than the minimum size (400) recommended based on statistical considerations (Fore *et al.*, in press). Using a bootstrap resampling algorithm to conduct repetitive IBI analyses of a large data set from Ohio streams, Fore *et al.* (in press) found that, 'IBI scores at sites for which less than 400 fish were collected were notably more variable.' Obtaining samples of 400 fish from extremely degraded sites is not always feasible, possible, or a wise use of the fishery resource. Sampling techniques for fish are always problematic. Our experience in this region indicates that backpack electroshocking is the most appropriate gear for the extremely complex habitats in the range of stream sizes we studied (water widths 3–12 m, depths 7–58 cm). Electroshocking has also been used with success for regional IBI studies in Ohio (Ohio EPA, 1989), southern Ontario (Steedman, 1988), and many other areas.

A fourth problem also relates to the lack of reference sites. Statistical power analyses indicate that the IBI may be used to discriminate among 5 distinct classes of biotic integrity. We were unable to detect significant physical differences between only two classes based on biotic integrity. However, our set of study sites may have been limited to too narrow a range of habitat and biotic conditions to detect gradients. The range of IBI scores we obtained (50–24 = 26) is slightly more than half as great as the possible range (60–12 = 48), and we were able to produce this wide a spread by adjusting our metric scoring criteria to fit the observed ranges. A more orthodox approach to measuring and assessing biotic integrity may have resulted in a narrower range of IBI scores and thus placed all, or nearly all, of our sites in one rather than two categories.

Sites with highest levels of relative biotic integrity consistently had woody debris densities between about 5000 and 20 000 m^2 km^{-2}, but woody debris was scarce (0 to 1000 m^2 km^{-2}) at sites with lowest relative IBI (Table 9). Large woody debris is an important structural component of warmwater fish habitats (Hickman, 1975, Hortle & Lake, 1983, Angermeier & Karr, 1984). In sandy warmwater streams, woody debris may be the locus for much of the invertebrate production (Benke *et al.*, 1985) and create higher levels of physical heterogeneity (Shields & Smith, 1992) and pool availability (Ebert *et al.*, 1991). Pool habitat and structural influence was also associated with highest levels of relative IBI when compared to sites with lowest scores. Pool habitat availability has been related to fish species richness in this region (Shields *et al.*, 1994, Shields & Hoover, 1991) and elsewhere (Evans & Noble, 1979; Schlosser, 1982; Meffe & Sheldon, 1988; Ebert *et al.*, 1991), and to catch per unit effort (Foltz, 1982). In disturbed, warmwater streams channel stabilization structures such as weirs (Cooper & Knight, 1987; Shields & Hoover, 1991; Edwards *et al.*, 1984), spur dikes (Carline & Klosiewski, 1985; Knight & Cooper, 1991b; Shields *et al.*, 1993) and other types of bank protection are often associated with higher levels of fish abundance, species richness, and species diversity due to the formation of stable pools and cover.

Subtle water quality influences may have confounded relationships between physical habitat quality and relative IBI scores. The scarcity of fish with diseases, parasites, or anomalies suggested that water quality was generally within tolerable limits, and with the exception of three slightly elevated specific conductance measurements (900–1200 μmhos cm^{-1}), the limited data we collected indicated no water quality problems. However, other studies of some of the same streams have reported elevated (3000 to 5000 mg l^{-1}) suspended solids concentrations during and immediately after storm events (Cooper & Knight, 1991; Rebich, 1993). Studies of sites in two of the watersheds found that and stream bed sediments were sometimes contaminated with arsenic, mercury, and residues from long-lived organochlorine pesticides that were applied historically, but are no longer used (Knight & Cooper, 1991a; Cooper *et al.*, 1992). One of these studies revealed that fish tissues contained concentrations of organochlorine insecticides, and on occasion, currently used insecticides, that were high relative to adjacent water and sediments, suggesting bioaccumulation (Knight & Cooper, 1991a). Effects of these residues on fish assemblage characteristics that are the IBI metrics are unknown. Invertebrate bioassays have also revealed that bed sediments from some of the streams we studied contain toxic levels of contaminants (Knight *et al.*, 1994).

Conclusions

Although the streams we studied have been severely degraded, they still harbor a large number of fish species (65), albeit in fragmented ranges. This fact, coupled with the low intensity of land use in much of the study area, indicates significant potential for restoration of stream corridor ecosystems. Complete restoration of pre-European settlement conditions is not practical, and therefore decisions have to be made regarding the selection of habitat characteristics for treatment. Because our formulation of the index of biotic integrity was indicative of only the grossest differences in habitat quality, it can be used as only a general guide for restoration decisions. The weak relationships between metrics of physical habitat quality and relative IBI scores may be due to large temporal variations in biotic integrity typical of degraded habitats, particularly warmwater streams that have been transformed by accelerated erosion and sedimentation. Alternatively, water quality degradation, which we did not measure, may have confounded relationships among physical and biological variables.

Regional application of the IBI as an assessment tool in a landscape with widespread physical degradation of stream habitats must overcome several obstacles. Among these are the lack of suitable reference collections for setting metric scoring criteria, large temporal variation in IBI scores at a given site, and small numbers of fish per collection leading to lower confidence levels for IBI scores. Finally, the scarcity of reference sites may also make it difficult to establish gradients of biotic integrity and physical habitat quality.

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References

Angermeier, P. L. & J. R. Karr, 1986. Applying an index of biotic integrity based on stream-fish communities: considerations in sampling and interpretation. N. am. J. fish. Mgmt 6: 418–429.

Angermeier, P. L. & J. R. Karr, 1984. Relationships between woody debris and fish habitat in a small warmwater stream. Trans. am. Fish. Soc. 113: 716–726.

Angermeier, P. L. & I. J. Schlosser, 1987. Assessing biotic integrity of the fish community in a small Illinois stream. N. am. J. fish. Mgmt 7: 331–338.

Benke, A. C., R. L. Henry, D. M. Gillespie & R. J. Hunter, 1985. Importance of snag habitat for animal production in southeastern streams. Fisheries 10: 8–13.

Bennett, D. H. & T. R. Fisher, 1989. Use of the index of biotic integrity to assess the impact of land management activities on low order streams in northern Idaho. Research Technical Completion Report 14-08-0001-G1419-06, Idaho Water Resources Research Institute, University of Idaho, Moscow.

Berkman, H. E., C. F. Rabeni & T. P. Boyle, 1986. Biomonitors of steam quality in agricultural areas: fish versus invertebrates. Envir. Mgmt 10: 413–419.

Berkman, H. E., C. F. Rabeni, 1987. Effect of siltation on stream fish communities. Envir. Biol. Fish. 18: 285–294.

Bramblett, R. G. & K. D. Fausch, 1991. Variable fish communities and the index of biotic integrity in a western Great Plains river Trans am. Fish. Soc 120: 752–769

Carline, R F & S. P. Klosiewski, 1985 Responses of fish populations to mitigation structures in two small channelized streams in Ohio. N. Am J. Fish. Mgmt 5: 1–11

Cooper, C. M. & S. S. Knight, 1987. Fisheries in man-made pools below grade-control structures and in naturally occurring scour holes of unstable streams J Soil Water Conserv. 42: 370–373.

Cooper, C. M & S. S. Knight, 1991. Water quality cycles in two hill land streams subjected to natural, municipal, and non-point agricultural stresses in the Yazoo Basin of Mississippi, USA (1985–1987) Verh int Ver Limnol. 24: 1654–1663

Cooper, C. M., P. G. McCoy & S S Knight, 1992 Evaluation of the Coldwater River, January, 1989 through June, 1991. National Sedimentation Laboratory Research Report No 1, Interim Report for The Demonstration Erosion Control (DEC) Task Force. Oxford, Mississippi: United States Department of Agriculture, Agricultural Research Service, National Sedimentation Laboratory, 32 pp

Crumby, W. D., M A Webb, F J. Bulow & H J Cathey, 1990 Changes in biotic integrity of a river in north-central Tennessee Trans am Fish Soc 119: 885–893.

Ebert, D. J , T A Nelson & J L Kershner,1991 A soil-based assessment of stream fish habitats in coastal plains streams. Warmwater Fisheries Symposium I, General Technical Report RM-207, USDA Forest Service, Washington, D.C , 217–224.

Edwards, C J., B L Griswold, R. A. Tubb, E C. Weber & L. C Woods, 1984 Mitigating effects of artificial riffles and pools on the fauna of a channelized warmwater stream. N. Am. J Fish. Mgmt 4: 194–203.

Emerson, J W, 1971 Channelization A case study. Science 173 325–326.

Evans, J. W & R L Noble, 1979 The longitudinal Am Midl Nat. 101: 333–343.

Fausch, K. D , J. R. Karr & P. R Yant, 1984. Regional application of an index of biotic integrity based on stream fish communities Trans. am. Fish. Soc 113: 39–55.

Fausch, K. D , J Lyons, J R Karr & P L Angermeier, 1990 Fish communities as indicators of environmental degradation. Am. Fish Soc Symp. 8: 123–144.

Foltz, J W., 1982. Fish species diversity and abundance in relation to stream habitat characteristics. Proceedings of the Thirty-Sixth Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 305–311.

Fore, L S., J R. Karr & L L Conquest. In press. Statistical properties of an index of biological integrity used to evaluate water resources Can. J. Fish aquat. Sci

Frissell, C A , W J Liss, C E. Warren & M. D Hurley, 1986. A hierarchical framework for stream habitat classification. viewing streams in a watershed context Envir Mgmt 10: 199–214

Galay, V J , October 1983 Causes of river bed degradation Wat Resour. Res. 19 1057–1090.

Gorman, O T. & J R Karr, 1978. Habitat structure and stream fish communities Ecology 59 507–515.

Grissinger, E. H., A. J. Bowie & J. B Murphey. 1991. Goodwin Creek Bank Instability and Sediment Yield. Proceedings of the Fifth Federal Interagency Sedimentation Conference (FIASC), Las Vegas, Nevada, 1991 Federal Energy Regulatory Commission, PS32–PS39

Grissinger, E H & J. B. Murphey, 1986 River sedimentation (volume III): bank and bed adjustments in a Yazoo bluffline tributary In S. Y Wang, H W. Shen & L Z Ding, (eds), Third International Symposium on River Sedimentation, The University of Mississippi, University, Mississippi 1003–1012.

Harvey, M D & C C Watson, 1986 Fluvial processes and morphological thresholds in incised channel restoration Water Resour Bull Am. Water Res. Ass. 22 359–368

Hickman, G. D , 1975 Value of instream cover to the fish populations of Middle Fabius River, Missouri. Aquatic Series No 14

Hilgard, E. W., 1860 Report on the geology and agriculture of the State of Mississippi. Miss. State Geological Survey, Jackson, 391 pp

Hortle, K G & P S Lake, 1982 Macroinvertebrate assemblages in channelized and unchannelized sections of the Bunyip River, Victoria Aust. J mar. Freshwat Res. 33: 1071–1082.

Hortle, K G. & P S Lake, 1983. Fish of channelized and unchannelized sections of the Bunyip River, Victoria. Aust. J mar Freshwat Res 34 441–450

Hughes, R M & J R. Gammon, 1987 Longitudinal changes in fish assemblages and water quality in the Willamette River, Oregon Trans am. Fish Soc 116 196–209.

Jester, D. B , A A. Echelle, W. J. Matthews, J. Pigg, C. M. Scott & K. D. Collins, 1992 The fishes of Oklahoma, their gross habitats, and their tolerance of degradation in water quality and habitat. Proc Okla Acad. Sci. 72: 7–19.

Joseph, M W, 1993. An intensive survey of the Edwards River and Pope Creek. Report No IEPA/WPC/93-141, Illinois Environmental Protection Agency, Bureau of Water, Springfield.

Kanehl, P. & J. Lyons, 1992. Impacts of in-stream sand and gravel mining on stream habitat and fish communities, including a survey on the Big Rib River, Marathon County, Wisconsin. Wisconsin Department of Natural Resources, Research Report 155

Karr, J. R , 1987 Biological monitoring and environmental assessment. a conceptual framework. Envir Mgmt 11 249–256.

Karr, J. R., 1991. Biological integrity: A long-neglected aspect of water resource management. Ecol Appl. 1: 66–84

Karr, J R, 1992. Measuring biological integrity. lessons from streams. Ecological Integrity and the Management of Ecosystems. S. J Woodley, G. Francis & J. Kay (eds), Ann Arbor, Michigan CRC Press 79–99

Karr, J. R , 1993 Defining and assessing ecological integrity beyond water quality. Envir. Toxicol. Chem 12: 1521–1531.

Karr, J. R , K. D. Fausch, P. L. Angermeier, P R Yant & I J Schlosser, 1986 Assessing biological integrity in running waters. a method and its rationale. Illinois Natural History Survey, Special Publication No 5 Illinois Natural History Survey, Champaign

Karr, J R & B L. Kerans, 1992 Components of biological integrity: their definition and use in development of an invertebrate IBI, EPA 905/R-92/003 Proceedings of the 1991 Midwest Pollution Control Biologists Meeting, Environmental Indicators Measurement and Assessment Endpoints. editors T P. Simon & W S Davis, 1–16 Chicago, Illinois: U.S Environmental Protection Agency, Region V, Environmental Sciences Division.

Karr, J R , P R Yant, K D Fausch & I J. Schlosser, 1984. Evaluation of an index of biotic integrity: temporal variability and regional application in the Midwest Environmental Research Brief, EPA-600/3-83-006 U S Environmental Protection Agency, Corvallis, Oregon

Karr, J R., P. R Yant, K. D Fausch & I. J. Schlosser, 1987. Spatial and temporal variability of the index of biotic integrity in three midwestern streams. Trans am Fish Soc 116: 1–11.

Kesel, R. H. & E G Yodis, 1992 Some effects of human modifications on sand-bed channels in southwestern Mississippi, USA Envir. Geol. Water Sci 20 93–104.

Killgore, K. J. & N H Douglas, 1988. Effects of surface water withdrawal on fishes in rivers of northeast Louisiana. Technical Report EL-88-2, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Knight, S. S & C M Cooper, 1987. Fishes of Otoucalofa Creek, Mississippi prior to major channel modifications J. Miss. Acad Sci 22: 31–38.

Knight, S S. & C M. Cooper, 1990 Fishes of Hotophia Creek, Mississippi J. Miss acad. Sci 1–12.

Knight, S S & C M. Cooper, 1991a Distribution of insecticides and metals in a mixed agricultural watershed. In Proceedings for the Environmentally Sound Agriculture Conference, 2 681–688

Knight, S S. & C. M Cooper, 1991b Effects of bank protection on stream fishes. Proceedings of the Fifth Federal Interagency Sedimentation Conference (FIASC) Federal Energy Regulatory Commission, Washington, D.C.: 13-34–13-39

Knight, S S, C. M. Cooper & S. Testa, 1994 Use of Hyallela Azteca (Saussure) mortality to complement ongoing water quality research. Bulletin of the North American Benthological Society 11(1)· 153.

Leonard, P M. & D. J Orth, 1986. Application and testing of an index of biotic integrity in small, coolwater streams Trans am Fish Soc. 115. 401–414

Lyons, J , 1992 Using the index of biotic integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. General Technical Report NC-149 51 St Paul, Minnesota North Central Forest Experiment Station, Forest Service, U S Department of Agriculture

Meffe, G. K. & A. L. Sheldon, 1988. The influence of habitat structure on fish assemblage composition in southeastern blackwater streams Am Midl Nat. 120. 225–240

Menzel, B. W , J. B Barnum & L. M Antosch, 1984. Ecological alterations of Iowa prairie-agricultural streams Iowa State J. Res. 59. 5–30.

Miller, D L., P. M. Leonard, R. M. Hughes, J R Karr, P. B Moyle, L. H Schrader, B. A Thompson, R A Daniels, K D Fausch, G. A. Fitzhugh, J. R Gammon, D B Halliwell, P. L. Angermeier & D Orth, 1988. Regional applications of an index of biotic integrity for use in water resource management. Fisheries 13. 12–20.

Ohio Environmental Protection Agency, 1989 Biological criteria for the protection of aquatic life. volume III Standardized biological field sampling and laboratory methods for assessing fish and macroinvertebrate communities Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, Ohio.

Osborne, L. L , B. Dickson, M Ebbers, R Ford, J Lyons, D Kline, E Rankin, D. Ross, R. Sauer, P. Seelbach, C. Speas, T. Stefanavage, J Waite & S Walker, 1991. Stream habitat assessment programs in States of the AFS North Central Division Fisheries 16: 28–35.

Patrick, D. M., L Mao & S. T Ross, 1991 Accelerated erosion in the Bayou Pierre Basin, southwest Mississippi· characterization and causes. In B. J. Daniel, (ed), Twenty-first Mississippi Water Resources Conference, Mississippi State University· Water Resources Research Institute (Mississippi State University) 36–44

Petersen, R C Jr., 1992 The RCE a riparian, channel, and environmental inventory for small streams in the agricultural landscape. Freshwat Biol. 27· 295–306

Piest, R F., L S. Elliot & R G. Spomer, 1977. Erosion of the Tarkio drainage system, 1845–1976 Trans am Soc. Agric. Engin 20 485–488.

Plafkin, J. L., M. T Barbour, K D Porter, S. K. Gross & R M. Hughes, 1989. Rapid bioassessment protocols for use in streams and rivers benthic macroinvertebrates and fish. EPA/444/4-89-001 U S Environmental Protection Agency, Washington, D.C

Rankin, E T. & C O. Yoder, 1990 The nature of sampling variability in the index on biotic integrity (IBI) in Ohio streams. Proceedings of the 1990 Midwest Pollution Control Biologists Meeting. editor W P Davis, U.S. Environmental Protection Agency

Rebich, R A , 1993 Preliminary summaries and trend analyses of stream discharge and sediment data for the Yazoo River basin demonstration erosion control project, north-central Mississippi, July 1985 through September 1991. U.S Geological Survey, Water Resources Investigation Report 93-4068, Jackson, Miss

Robison, H W. & T. J Buchanan, 1988. Fishes of Arkansas. University of Arkansas Press, Fayetteville, 536 pp.

Ross, S T & W. M Brenneman, 1991. Distribution of freshwater fishes in Mississippi. Completion report, Dingell-Johnson project F-69, Mississippi Dept Wildlife, Fisheries, and Parks, Bureau of Fisheries and Wildlife, Jackson, 548 pp

Saylor, C. F & S. Ahlstedt, 1990. Application of index of biotic integrity (IBI) to fixed station water quality monitoring sites. TVA/WR/AV-90/12 Tennessee Valley Authority, Norris, Tennessee

Schlosser, I J., 1982 Fish community structure and function along two habitat gradients in a headwater stream. Ecol. Monogr. 52: 395–414.

Shields, F D , Jr., S. S. Knight & C. M. Cooper, 1994. Effects of channel incision on base flow stream habitats and fishes. Envir Mgmt 18: 43–57.

Shields, F. D , Jr , C M. Cooper & S. S. Knight, 1993. Initial habitat response to incised channel rehabilitation. Aquat. Conserv : Mar Freshwat. Systems 3: 93–103

Shields, F D , Jr & J J Hoover, 1991 Effects of channel restabilization on habitat diversity, Twentymile Creek, Mississippi Regulated Rivers: Res. Mgmt 6: 163–181

Shields, F D , Jr. & R H. Smith, 1992 Effects of large woody debris removal on physical characteristics of a sand-bed river. Aquat. Conserv Mar. Freshwat Ecosystems 145–163.

Simon, A., 1989 The discharge of sediment in channelized alluvial streams. Wat. Resour Bull 25. 1177–1188.

Simon, A. & C H Robbins, 1987. Man-induced gradient adjustment of the South Fork Forked Deer River, West Tennessee. Envir. Geol. Water Sci. 9: 109–118.

Slack, L. J., 1992. Water-quality and bottom-material-chemistry data for the Yazoo River basin demonstration erosion control project, north-central Mississippi, February 1988–September 1991. U.S. Geological Survey Open-File Report 92-469 197 Jackson, Mississippi U.S Geological Survey.

Steedman, R. J , 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. Can J. Fish. aquat. Sci. 45. 492–501.

Suttkus, R D., 1991 Notropis ranfinesquei, a new cyprinid fish from the Yazoo River system in Mississippi. bulletin of the Alabama Museum of Natural History 10· 1–9.

Whitten, C B & D M Patrick, 1981 Engineering geology and geomorphology of streambank erosion (Report 2. Yazoo River Basin Uplands, Mississippi) Technical Report GL-79-7, Report 2 of a Series